Is agricultural management promoting carbon sequestration better for the climate?

Comparing the carbon footprint of intervention management and conventional management using an LCA approach

TOM BRILIOTH 2024
MVEM14 EXAMENSARBETE FÖR MASTEREXAMEN 30 HP
MILJÖVETENSKAP | LUNDS UNIVERSITET

Is agricultural management promoting carbon sequestration better for the climate?

Comparing the carbon footprint of intervention management and conventional management using an LCA approach

Tom Brilioth

2024

LUNDS UNIVERSITET

Tom Brilioth

Master thesis

MVEM14 Examensarbete för masterexamen 30 hp

Supervisors: Raül López i Losada & Katarina Hedlund, Centre for Environmental and Climate Science, Lund University

CEC - Centre for Environmental and Climate Science
Lund University
Lund 2024

Cover photo: Gregory, E. (2016). *Field-agriculture-farm-grass* [Photograph]. Wikimedia Commons. https://commons.wikimedia.org/wiki/File:Field-agriculture-farm-grass_%2823697661484%29.jpg

Abstract

Improved management of agricultural soils to sequester carbon from the atmosphere is considered an important climate mitigation measure. Intensive agriculture often leads to degradation of soil organic carbon (SOC), contributing to climate change and loss of soil ecosystem services. Interventions in agricultural management can halt or reverse the loss of SOC.

Annual SOC change rates for intervention and conventional management were taken from a review on long-term time series data. The purpose of this study is to compare the GHG emissions of management scenarios from a life cycle perspective. The intervention scenario differs from the conventional scenario by using manure to replace inorganic fertiliser, having reduced tillage, and leaving crop residues in the field. Production functions were used to describe the relationship between SOC, nutrient inputs and yields. Using life cycle assessment (LCA) methods, the greenhouse gas (GHG) emissions of each scenario are estimated based on regional data combined with a LCA database. The functional unit is yields converted into cereal units for a full crop rotation, after 20 years of modelled SOC changes.

This study shows that an intervention management has substantially lower GHG emissions compared to the conventional scenario. Much of the total difference between scenarios is attributed to the difference in SOC changes, which are positive in the intervention scenario and negative in the conventional. Soil N_2O emissions were high in both scenarios but were lower in the intervention scenario because of lower inorganic fertiliser application. However, the method for assessing soil N_2O emissions produces rougher estimations than the method for SOC changes, and the full effects of management on N_2O emissions has not been quantified in this study. Higher SOC in the intervention scenario also resulted in lower N input and higher yields, causing lower emissions per functional unit. Lower GHG emissions resulted also from replacing inorganic fertiliser with manure and reducing tillage. In conclusion, intervention agriculture is shown to perform substiantially better than conventional in terms of GHG emissions, but further research should investigare uncertainties in the result.

Contents

Introduction

Anthropogenic climate change has increasingly adverse effects on the planet and on society, requiring large efforts of mitigation and adaptation (IPCC, 2023). While the burning of fossil fuel is the primary driver, large greenhouse gas (GHG) emissions are also attributed to land use and land use change including agriculture and forestry (IPCC, 2019). The agricultural sector is one of the largest contributors to global warming (Tubiello et al., 2015). While agricultural production is needed to feed a growing population, its contribution to climate change, biodiversity loss and land degradation must be addressed to meet sustainability goals and provide food security (Foley et al., 2011). An important environmental aspect is the effect of agriculture on the soil, the foundational natural resource supporting agricultural production. Agricultural management can be improved to benefit carbon sequestration and other ecosystem services (Brady et al., 2015; Minasny et al., 2017). In the aims of Sweden and the European Union to reach climate neutrality by 2050, agriculture plays an important role as a potential carbon sink (European Commission, 2023).

Ecosystem services of agricultural soils

The complex organism communities of soil ecosystems provide ecosystem services essential to agricultural production, such as nutrient cycling and uptake, regulation of soil erosion, carbon sequestration and water flow as well as control of pests and diseases (Barrios, 2007). The intensification of agriculture has caused a decline of biodiversity in agricultural soils (Tsiafouli et al., 2015). One of the primary drivers of this biodiversity loss is the decline of soil organic carbon (SOC) (Gardi et al., 2013). An increase of SOC is correlated with beneficial conditions for plant growth regarding soil structure, water-holding capacity and nutrient availability, meaning that SOC boosts soil productivity in terms of crop yields (Johnston et al., 2009; Oldfield et al., 2019). Changing SOC levels in soils can be considered a proxy for soil biodiversity and ecosystem services, which makes SOC valuable to farmers and to society (Brady et al., 2015).

There are large fluxes of carbon between the atmosphere and the biosphere, and soils have the largest carbon stocks on land (Schlesinger & Bernhardt, 2020). Therefore, it is important to consider the effects of soil management on carbon stocks in regard to climate change. Typically, the introduction of agriculture to previously untilled soils lead to a loss of organic carbon of about 30 percent (Davidson & Ackerman, 1993). A significant portion of agricultural GHG emissions can come from soil organic carbon (SOC) losses (Brandao et al., 2011; Joensuu et al., 2021). However, there is potential to halt soil carbon losses or even sequester large amounts of carbon in croplands (Lal, 2004; Zomer et al., 2017). Carbon sequestration in agricultural soils is considered an important climate mitigation measure, and a necessary tool to reach global climate goals (Lal, 2004; Minasny et al., 2017).

Management practices to halt or reverse soil carbon losses

Many different methods to sequester carbon in agricultural soils have been suggested and researched, including for example reduced or no tillage, applying organic fertilisers or crop residues, planting cover crops or using better crop rotations (Haddaway et al., 2015; Lal, 2004). Conventional tillage, being the norm in modern agricultural history, uses a mouldboard plough followed by harrowing to prepare the soil and combat weeds (Phillips et al., 1980). This disturbance in the soil can stimulate decomposition of organic matter, leading to soil carbon losses. Reducing the disturbance by tilling less or not at all could then have the potential to reduce SOC losses or contribute to carbon sequestration (Alvarez, 2005). The potential of reduced or no tillage to mitigate climate change has been questioned (Baker et al., 2007; VandenBygaart, 2016). Some evidence suggests that the key is to combine reduced tillage with other practices to achieve carbon sequestration (Blanco-Canqui, 2021).

An increased amount of nitrogen through fertilisation can promote carbon sequestration, if crop residues are returned to the soil (Alvarez, 2005). However, despite the effects of conventional fertilisation, agricultural soils across the world have continuously been degraded and SOC stocks have declined (Khan et al., 2007). Applying organic fertilisers, i.e. manure from animals, can have a greater effect than inorganic fertilisers in increasing SOC, since the manure itself contains organic material (Blair et al., 2006).

The planting of cover crops, planted in between seasons not primarily for harvest but for their benefits to the soil, can halt erosion, improve soil structure, and add biomass, including carbon, and nitrogen, increasing carbon sequestration and reducing N leaching (Abdalla et al., 2019; Bai et al., 2019). Leaving crop residues in the fields does some of the same good as cover crops or manure by adding organic material to the soils (Bolinder et al., 2020). According to Prechsl et al. (2017), the planting of cover crops could also have negative side effects, causing lower yields and higher soil N_2O emissions.

In addition to the loss of SOC to the atmosphere, the agricultural sector contributes substantially to the emissions of N_2O and CH_4 (IPCC, 2019). N_2O is a long-lived GHG with a higher effective radiative forcing (ERF) than CO_2, while CH_4 is a short-lived GHG with an even higher ERF (Forster et al., 2021). In an LCA study, Brandao et al. (2011) found that both CO_2 and N_2O emissions from soil are significant contributors to agricultural climate impact. N_2O can be directly emitted from soils through denitrification under anoxic conditions. It can also be emitted indirectly via volatilized NH_3 (Pan et al., 2016). Methane emissions from soil are primarily associated with submerged soils and can be substantial for some types of agriculture, such as rice cultivation (Le Mer & Roger, 2001). While upland soils are typically a sink for methane, agricultural soils can be sources due to soil treatment and fertilisation (Cowan et al., 2021; Powlson et al., 1997). The reduction of tillage can reduce soil emissions of methane (Bayer et al., 2012).

The environmental impact of agricultural management is not limited to effects on the soil, GHG emissions can also result from all steps of the production from soil preparation to harvesting. In a report on life cycle inventories for ecoinvent, Nemecek and Kägi (2007) describe both agricultural infrastructures, including buildings, machinery and field processes, and agricultural inputs, including inorganic and organic fertilisers, pesticides, seed and drying of grain and grass. For example, the production of inorganic nitrogen fertiliser through the Haber-Bosch process results in large GHG emissions, a significant contribution to the total agricultural emissions (Menegat et al., 2022). On-farm machine/fuel use, fertiliser inputs have been identified in agricultural LCAs as major sources of greenhouse gas emissions (Aguilera et al., 2015; Camargo et al., 2013; Meisterling et al., 2009). Direct soil emissions of CO_2 and N_2O are also significant but varies a lot between these studies. As described below, the method of accounting for soil related emissions can vary and impacts the result.

Accounting for soil carbon sequestration

To determine which agricultural management practices are the most environmentally efficient, a method is needed to comprehensively account for the effects of complex agricultural systems. Life cycle assessment (LCA) has frequently been used for this purpose as it is a well-established method for assessing all processes and environmental impacts of a production system (Finnveden et al., 2009). LCA considers the environmental impact of a product in relation to its function, such as the amount of product produced, which indicates the efficiency of the production system. However, there are specific challenges in conducting LCA for agricultural systems, and no consensus has been reached on how to account for SOC changes (Goglio et al., 2015).

Soils are complex eco-systems and take time to stabilize after changes in management, including the SOC content (Ludwig et al., 2011). Common methods for accounting for SOC changes in agriculture have been criticized by Sanderman and Baldock (2010) for using short-term or single point in time measurements. Such methods assume that SOC levels start at a stable baseline, an equilibrium, which might not be the case. The use of response ratios, comparing the SOC contents soil under a certain management compared to a conventional or baseline reference, have been criticised in favour of using annual change rate (Sanderman & Baldock, 2010). Joensuu et al. (2021) found that the assumptions on the state of SOC stocks, considering historical land use and management, and the time horizon significantly affects the assessment of SOC changes in an agricultural LCA. Changes in SOC typically progress over a time period of decades, before stabilising (Johnston et al., 2009). This leads to limited predictive power of methods based on short term or single point measurements, and overestimating carbon sequestration in some scenarios. A full LCA of a cropping system should also account for emissions of N_2O from the soil, however these emissions vary based on many factors and different assessment methods can have varying results (P. Goglio et al., 2018).

Haddaway et al. (2016) published a review protocol, aimed to address the issues raised by Sanderman and Baldock (2010) and assess influence on SOC by management interventions based on time series data. Using long-term, high-quality data to assess how SOC changes from different management practices is an important step towards determining which agricultural methods should be used to reduce the climate impact. The protocol describes the method for a review of academic and grey literature meeting the pre-defined selection criteria: the selected studies should be from a warm temperate or snow climate zones where wheat can grow, investigate different management practices, or interventions, (including tillage, fertilizer, crop rotations and amendments) and must measure soil C (Haddaway et al., 2016). Studies included must present time series data over a period of at least 30 years (Haddaway et al., 2016). An unpublished review based on the protocol by Haddaway et al. (2016) has been conducted by researchers at Lund university, resulting in annual SOC change rates for different sets of management.

Purpose and research questions

The purpose of this thesis is to assess the climate impact of agricultural management methods that improve SOC content in agricultural soils, compared to conventional methods, based on long-term time series data on SOC changes. This includes the greenhouse gas emissions of all inputs to and processes on the farm up until the crop is harvested. The long-term data on SOC changes will determine what agricultural methods are promoting carbon sequestration and will be included in the study. The emissions from the agricultural management will be compared to the carbon losses or sequestration of the soil, to conclude whether the results support the notion that these interventions for the soil are beneficial for the climate, or not.

The research questions are:

- What is the life cycle performance in terms of GHG emissions of agricultural management with interventions to improve SOC in crop production, compared to conventional management?
- What aspects and processes of the crop production differ significantly between the scenarios in terms of GHG emissions, and which contribute the most to the total GHG emissions?

Limitations

The study is limited to crop cultivation in the Swedish agricultural region of Götalands södra slättbygder (GSS) (Jordbruksverket, 2020). Generalization of the results to other regions should be done carefully. Furthermore, the "Goal and scope" section of the method defines the limits of the crop production system. For example, the processing of harvested crops into food products such as flour is not included. The temporal limit of the assessment is 20 years into the future.

Methods and materials

The research questions of the thesis requires two main tasks to be carried out: first to construct scenarios that describe intervention management and conventional management within the scope of the study, and second to assess the life cycle performance in terms of greenhouse gas (GHG) emissions for both scenarios. The first segment of the method defines the two scenarios, describes the basis of the SOC changes in each scenario and how yields and nutrient inputs were calculated based on the SOC content of the soil. The second part of the method describes in detail how the method of life cycle assessment (LCA) was used to complete the second task, assessing the GHG emissions of the scenarios. An advantage of LCA is the ability to describe a complex system using only a few metrics of environmental impact. Combining all of the methods described here is what produces the results of this study.

Agricultural management scenarios

Two scenarios were constructed and compared, one intervention scenario with management practices to improve SOC, and one conventional scenario with management practices typical for GSS. The conventional scenario is mostly based on data from AgriWise, a planning tool for farmers maintained by the Swedish Board of Agriculture and the Swedish University of Agricultural Sciences based on expert and professional knowledge (AgriWise, 2020). GSS is characterized by a high prevalence of intensive crop cultivation due to beneficial climatic and soil conditions, which has also led to a decline in SOC compared to neighbouring regions (Brady et al., 2019). The climate is warm temperate, fully humid according to the Köppen-Geiger climate classification (Kottek et al., 2006). The nutrient application from organic fertilisers is lower than in other parts of Sweden while inorganic fertiliser inputs are high (SCB, 2023). The four-year crop rotation is winter wheat, winter rape seed, spring barley and sugar beet. The yield for each crop refers to the harvested part of the crop, excluding for example straw from the grain and leaves of the sugar beets. Wheat and barley are harvested at a 14 % moisture content, rapeseed at 11,5 % moisture content.

Table 1
Definition of management scenarios. Relevant types of management are those whose effect on SOC changes are investigated in the review (Haddaway et al., 2016).

MANAGEMENT	INTERVENTION SCENARIO	CONVENTIONAL SCENARIO
Crop rotation	Winter wheat – Winter rapeseed – Spring barley – Sugar beet	Winter wheat – Winter rapeseed – Spring barley – Sugar beet
Fertiliser application	Organic (30 kg N ha^{-1}) and inorganic	Inorganic
Pesticide application	Conventional	Conventional
Tillage	Reduced, no ploughing	Conventional
Crop residue management	Left in field	Removed from field

The intervention scenario differs from the conventional scenario in terms of organic and inorganic fertiliser inputs, tillage, and crop residue management. In this thesis, I use soil carbon change rates that are the result of an unpublished review in accordance with the review protocol by Haddaway et al. (2016). The data is currently not available in any publication and was received from one of the authors of this review, Raül López i Losada, a PhD student at Lund University and co-supervisor of this thesis. The data used from the review define the 95 % confidence interval of soil carbon change rates for scenarios with different number of interventions (R. López i Losada, 2024, personal communication). Only when all four types of interventions are applied (crop rotation, reduced tillage, organic fertiliser and leaving crop residues), the entire confidence interval is above 0, and there is a significant positive change, i.e. carbon sequestration. Based on this, the intervention scenario is defined as having all four types, while the conventional scenario only has one (crop rotation). In the review, having a crop rotation that differs from monoculture is considered an intervention, but having a crop rotation with several crops is common practice even in intensive conventional crop cultivation in Sweden.

The SOC contents and change rates are limited to the top 30 cm of the soil. The baseline SOC content in the region is assumed to be 1.71 % (Brady et al., 2019). The 95 % confidence interval for the change rate is -0.405±0.145 for the conventional scenario and 0.17±0.14 for the intervention scenario (R. López i Losada, 2024, personal communication). These values are the only information retrieved from the unpublished review, and could therefore be used to replicate this study. The relative changes of SOC were converted to changes in kg of carbon based on average soil characteristics in GSS (Brady et al., 2019). The positive or negative change of carbon was then converted to sequestration or emissions of CO_2-equivalents.

Table 2
Selection criteria for studies included in the unpublished review which SOC change rates are based on, according to the review protocol by Haddaway et al. (2016). The actual review may differ slightly from the review protocol.

Criteria	Description
Relevant populations	Arable soils in agricultural regions from the warm temperate climate zone and the snow climate zone.
Relevant interventions	Agricultural management practices relating to: different types, methods or amounts of **amendments** (including manure, crop residues, green manure, and more) or **fertilizer additions**, **tillage intensity** (no tillage/direct drill, reduced/conservation tillage, rotational/occasional tillage, conventional tillage, subsoiling); and **crop rotations** (monocultures, different crop sequences and rotation lengths, legumes, fallow, energy crops, annuals, perennials).
Relevant outcomes	Soil C measures, including SOC, total organic carbon (TOC), total carbon (TC), or soil organic matter (SOM).
Relevant study types	Interventions in the studies must have taken place over at least 30 years. At least three measurements must be made in this time.

Production functions and fertiliser input

To determine the use of nitrogen (N) fertiliser for different scenarios, production functions were used according to Brady et al. (2019). The functions estimate the relationship between SOC content, N input and yields, using coefficients based on long term experimental field studies. The coefficients in Brady et al. (2019) were calibrated according to instructions from Mark Brady, researcher at the Swedish University of Agricultural Sciences and lead author of the paper, to fit expected yields and practices of the region, using regional data from Agriwise (Table A1 in Appendix) (M. Brady, 2024, personal communication). It is assumed that the N/yield ratio for a certain SOC level is held at the optimal level, i.e. that the farmer determines the N input to maximize profits based on crop prices and the cost of fertilisation. Therefore, N input levels can be predicted from SOC change rates with the limitation that prices and costs are assumed to be constant. The values for used crop prices and fertilisation costs are from Brady et al. (2019) based on AgriWise data for 2020. Inputs of phosphorus (P) and potassium (K) were assumed to be linearly dependent on N input, based on Liebig's law of the minimum, and determined using ratios from AgriWise.

In the conventional scenario, 100 % of fertiliser needs are assumed to be met by inorganic fertilisers. In the intervention scenario, inorganic N fertiliser is partially replaced by manure equalling 30 kg of plant-available N. The inorganic P and K fertilisers are also replaced by manure based on the amount of manure corresponding to 30 kg of N. The concentration of N, P and K in manure was calculated based on data from Swedish Statistics which describes the total amount of manure applied in

GSS in terms of total weight as well as in kg of N, P and K (SCB, 2023). This data is therefore representative of the mix of different kinds of manure, i.e. in different forms from different animals, that is typically used in GSS.

Life cycle assessment

According to the ISO standard for LCA principles and framework, an LCA addresses the environmental aspects and potential environmental impacts of a product or service through its entire life cycle (Svenska institutet för standarder [SIS], 2006a). LCA has frequently been used to assess the impacts of agriculture (Goglio et al., 2015). The purpose of an LCA can for example be to identify how to improve the environmental performance of a product, or to advice decision-makers (SIS, 2006a). The ISO framework consists of four phases, which are executed in an iterative process: Goal and scope definition, inventory analysis, impact assessment and interpretation (SIS, 2006a).

Table 3
Summary of all inputs and processes of the crop production included in the life cycle inventory.

INPUTS	FIELD PROCESSES	POST HARVEST
Seed cultivation **Inorganic fertilisers; Pesticides** - Raw materials - Production, energy - Packaging - Transport **Manure** - Transport	**Sowing** **Tillage** **Fertiliser/manure and pesticide application** **Harvesting** - Production of machinery - Fuel use - Shed **SOC changes** **Soil N_2O emissions**	**Transport of crop** - Production of machinery - Fuel use

Goal and scope

There are two main categories of LCA, attributional and consequential, aimed at either accounting for the total environmental impacts of a product system or predicting the effect of changes to the system (Finnveden et al., 2009). The choice between these types is based on the goal of the LCA and has important consequences for how to set system boundaries and collect data (Tillman, 2000). The attributional modelling was applied in this study as it is appropriate when comparing the environmental impact of two systems, while consequential modelling may be used when investigating the effects of changes to one system (Finnveden et al., 2009).

The product system being studied is crop production in "Götalands södra slättbygder" (GSS), an agricultural region in southern Sweden. The system boundary should be consistent with the goal of the LCA, which is to compare different production systems with the same function (SIS, 2006b). It is not the main goal to determine the absolute and total climate impact of crop production, but rather any significant differences between production scenarios. Therefore, the most important environmental aspects to consider are those that can vary between scenarios. Including or excluding processes or inputs that are the same for all scenarios may affect the climate impact assessment of the system, but not the conclusions of the study. The main function of the system, and the function being considered in this study, is to produce food. The goal of this LCA is to compare the climate impact of the management scenarios, meaning that other relevant environmental impacts from agriculture such as eutrophication and toxicity are not considered. The LCA includes the effect on soil carbon in as well as other life cycle climate impacts of the crop production system (Table 3). The inputs of fertilisers and pesticides are considered, including the extraction of raw materials, transports, emissions from production and energy use.

The two production scenarios are assessed in the same way, with some variables changing between scenarios. Each growing season is assessed based on the characteristics of the scenario, crop type and the expected SOC content. One growing season starts after the harvest of the previous crop and ends with the harvest of the current crop. The impacts of each crop in the crop rotation are summarized to assess the impact of the system. Since the properties of the soil change over time, the inputs and impacts are projected 20 years into the future, to assess the effects of the soil carbon changes on other variables such as fertiliser use and yield.

Functional unit

LCA is a relative method, the environmental impacts are assessed in relation to the function that is produced (SIS, 2006a). For example, if one farm has twice the climate impact compared to another, but also produces twice the amount of food, their relative impact is the same. To quantify the function of a system under study, a functional unit must be chosen (SIS, 2006a). Agricultural systems are multifunctional, therefore it is difficult to condense the system function into a single functional unit and different options have been explored (Nemecek et al., 2011). The system in this study produces different crops: wheat, barley, rapeseed and sugar beet. Even when only the function of food production is considered, one kg of wheat grain is not identical to one kg of sugar beet. A method to overcome this issue is to convert different crops into a single unit called a cereal unit (CU), established by Brankatschk and Finkbeiner (2014). The method has previously been used in LCA to compare cropping systems including several different crops (Henryson et al., 2019; Prechsl et al., 2017). One CU corresponds to 1 kg of barley at 14 % moisture content, while other crops are converted based on their nutritional value for livestock (Brankatschk & Finkbeiner, 2014). The yields of the crops in this study were converted to CU based on conversion factors provided in the supplementary material of Brankatschk & Finkbeiner (2014). System functions other than food production, such as crop residues being used for animal feed or bioenergy, is not considered in this study.

Life cycle inventory

The life cycle inventory (LCI) was compiled by combining the use of an LCA database with region-specific data from farms in GSS. The ecoQuery website was used to access the ecoinvent database (version 3.8), a scientific global LCA database providing background data for the study (ecoinvent, 2021; Wernet et al., 2016). In ecoinvent, data on different agricultural processes are available, such as the sowing or ploughing, the production of inorganic N fertiliser or transportation using a tractor and trailer. For each of these so-called activities, the environmental impact per functional unit is determined. In this study, a library of relevant activities and their impact in terms of GHG emissions was compiled, while the amount of each activity included in the model was determined based mostly on regional data (table A2 in appendix). Few ecoinvent activities are based on data for Sweden specifically, meaning that German, Swiss or in some cases global data was used instead.

For the inputs of inorganic fertilisers, ecoinvent provides data for N, P and K for Sweden specifically, representing a mix of fertilisers based on national data on fertiliser consumption. These activities include raw material extraction, production, and transportation to customers. The input of packaging for fertilisers was calculated separately, based on the total weight of the most used fertilisers, representing >95 % of all inorganic fertiliser use. The production of manure was not allocated any environmental impact since it is considered a waste product from the animal industry. However, transportation using a tractor with a trailer as well as spreading of manure in fields was determined based on the amount of manure used, assuming a 60 km transport distance.

Data on average use of pesticides for different crops was retrieved from the Swedish statistics agency. Data specific to GSS was available for wheat, barley, and rape seed, but for sugar beet national data had to be used (SCB, 2022). Using ecoinvent data on unspecified pesticides, the climate impact was estimated. No difference between scenarios in pesticide use was assumed. Packaging of pesticides was not included since it was not practically possible to know the concentration of the active substance in the pesticides and therefore the actual amount to be packaged. It was also noted that even if the concentration was very low (e.g. 0,1 %), emissions from packaging would be negligible at the scale of other emissions, and therefore its omission does not impact the results.

In crop cultivation, various machines such as tractors are used for tillage, sowing, applying fertiliser and pesticides, harvesting, transports, and more. For this study, ecoinvent data on field operations was combined with Agriwise data on fuel consumption from farmers in GSS. Gustafsson and Johansson (2008) described different soil treatment regimes, conventional and with reduced tillage based on professional knowledge. Different regimes are adapted to different clay contents in the soil (Gustafsson & Johansson, 2008). Based on a map of clay content in Swedish agricultural soils, the most representative number for GSS was determined to be 20% (Piikki & Söderström, 2019). For the conventional and intervention scenarios in this study, plausible soil treatment regimens were created using corresponding ecoinvent activities, based on the descriptions by Gustafsson and Johansson (2008). Tillage was assumed to be the same for all crops. Data on the number of times fertilisation and application of pesticides are done were taken from Nilsson et al. (2023). The modelled amount of soil treatment activities as well as other field operations were adjusted to match Agriwise data on fuel consumption for different crops in GSS. The field operations considered are sowing, tillage, fertilisation (inorganic and manure), pesticide application and harvesting. Additionally, the transportation of the harvested crop to a storage site is included, as well as the transportation of manure to fields.

The impact of seeds for sowing was accounted for by subtracting from the yields, based on Agriwise values for how much seeds are needed per hectare. For sugar beet, where the seeds and what you harvest are different, ecoinvent data on how much sugar beet is produced per kg of sugar beet seed was used.

To quantify soil N_2O emissions in LCA, P. Goglio et al. (2018) recommended using the IPCC tier 2 approach. In this study, the direct N_2O emissions from the soil were calculated using a limited version of the IPCC tier 2 approach using disaggregated values for synthetic (inorganic) and organic N inputs, the limiting factor being that N inputs from crop residues was unknown (Hergoualc'h et al., 2019). The emission factors used were those for a wet climate (0,016 kg N2O-N kg N^{-1} for inorganic N, 0,006 kg N_2O-N kg N^{-1} for organic N and N mineralization from SOC losses). The indirect N_2O emissions were calculated using the IPCC tier 1 approach, using aggregated default values for volatilisation fractions and emission factors for wet climate (Hergoualc'h et al., 2019). The N_2O emissions were converted to CO_2-equivalents using characterization factors from Forster et al. (2021), which are the same for GWP20 and GWP100 (273 kg CO_2-eq kg N_2O^{-1}).

Impact assessment

LCA covers the potential environmental impacts of the assessed product, which typically includes a diversity of impacts (Finnveden et al., 2009). However, the impact categories should most importantly be chosen to address the goal of the study (SIS, 2006a). In this study, the only environmental aspect being considered is greenhouse gas emissions causing climate change. The used impact assessment method was ReCiPe, a model at the global scale based on current scientific understanding, including several different impact categories (Huijbregts et al., 2017). The midpoint characterisation factor for climate change used in ReCiPe is the widely used global warming potential (GWP) for different time horizons where short-lived greenhouse gases are given different weights. In this study, both 20-year and 100-year GWP was determined.

Results

Difference in GHG emissions

The life cycle performance of the intervention scenario in terms of greenhouse gas emisisons is better compared to conventional scenario (Figure 1). Based on mean values for the SOC changes, the net GWP100 per functional unit, including soil emissions of CO_2 and N_2O, is 49 % lower in the intervention scenario compared to the conventional. The total difference between the scenarios is 246 kg CO_2-eq, of which 174 kg, or 71 %, comes from the difference in SOC.

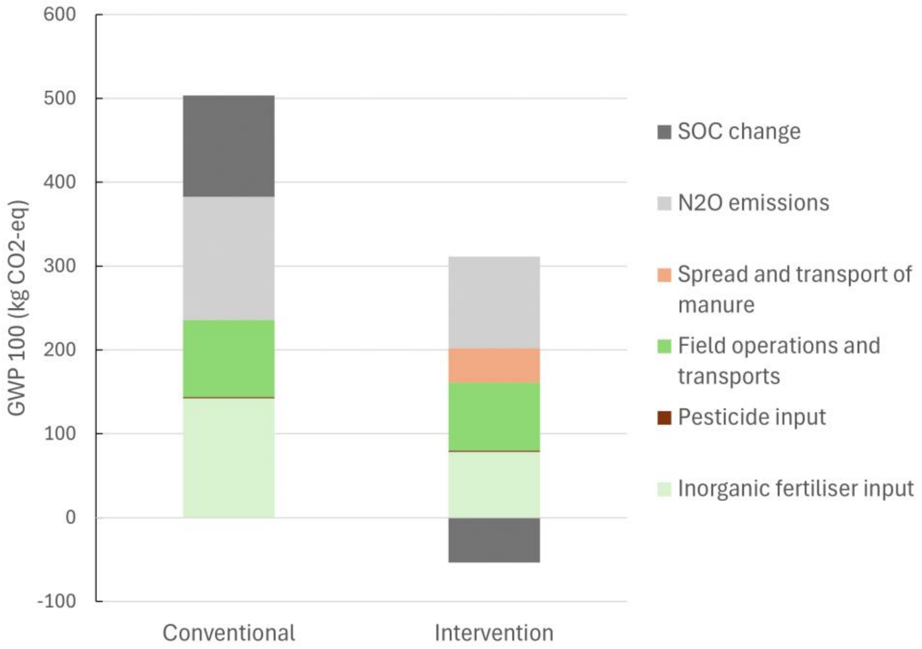

Figure 1. The total GWP100 per functional unit for each scenario, using mean values for SOC and corresponding variables. One full crop rotation, the last of the 20 year projection, is considered. The changes over time to SOC content, N input and yields derived from production functions are included. All inputs and processes described in table 3 are included.

Additionally, the difference in emissions can mostly be contributed to inorganic fertiliser inputs and soil N_2O emissions. The N_2O emissions from the soil are the single largest emission source in both scenarios, 29 % of total emissions in the conventional scenario and 31 % of the positive emissions (not considering SOC change) in the intervention scenario. The N_2O emissions are 26 % lower in the intervention scenario than in the conventional scenario. The GHG emissions from the management, i.e. the input of fertilisers, pesticides, and the use of machines for field operations and transports, are 14 % lower in the intervention scenario than in the conventional.

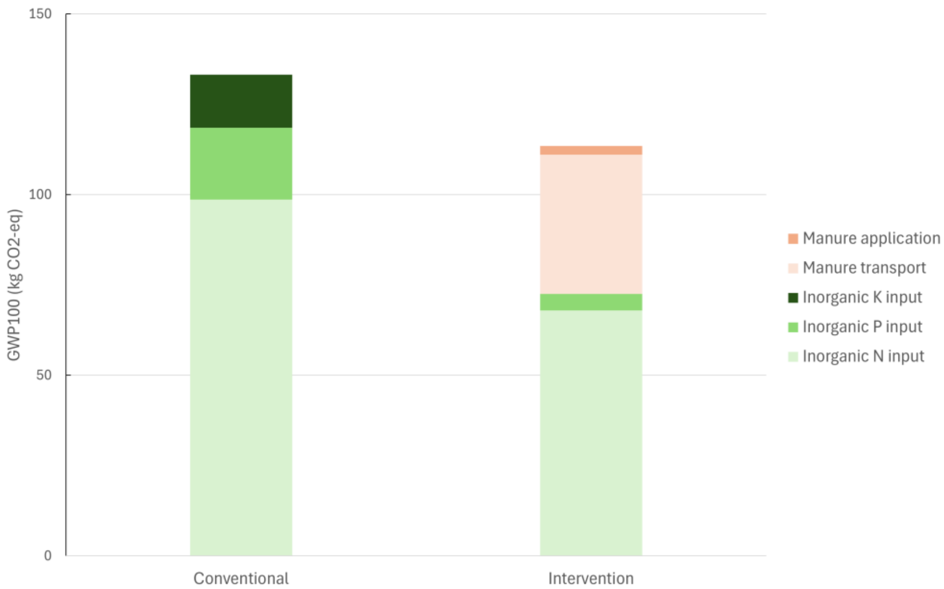

Figure 2 GWP100 per functional unit of fertiliser related emissions for each scenario. One full crop rotation, the last of the 20 year projection, is considered. The changes over time to SOC content, N input and yields derived from production functions are included.

The emissions from the production and use of fertilisers contribute substantially to the total emissions. The inorganic fertiliser input, including production, transport and packaging of said fertilisers account for 28 % of emissions in the conventional scenario and 22 % of positive emissions in the intervention scenario. The direct emissions of N_2O from the soil are also directly dependent on the input of N fertilisers, when calculated using the IPCC tier 1 approach. N is the nutrient with the highest input, thus the emissions from inorganic nitrogen fertiliser production are also high (Figure 2). The total N input is slightly lower in the intervention scenario due to the higher SOC, but most of the difference between inorganic N input between scenarios come from the replacement of inorganic fertiliser with manure.

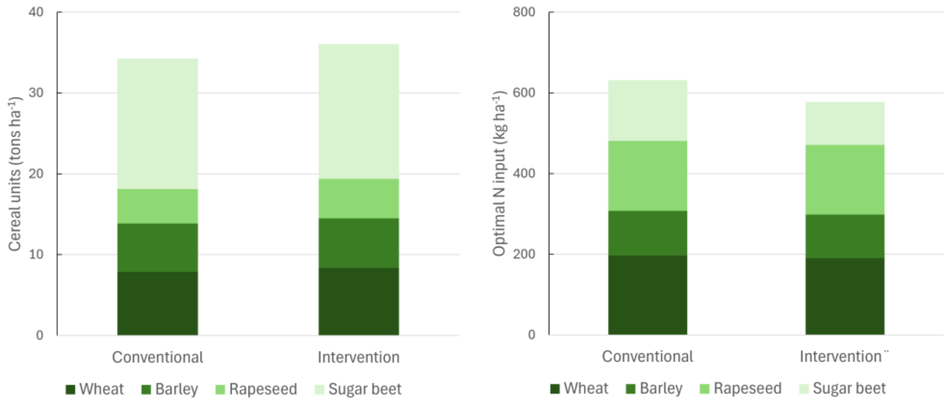

Figure 3 The contribution of different crops to the total yields (converted to CU) and total N input for each scenario. One full crop rotation, the last of the 20 year projection, is considered.

However, the large amounts of manure, ca 16 tons ha^{-1} yr^{-1} to get 30 kg plant-available N ha^{-1}, that must be transported leads to emissions that almost negate that reduction. Emissions from the transport and spread of manure amount to 11 % and 1 % respectively of total positive emissions in the intervention scenario. The amount of manure that is applied to reach 30 kg ha^{-1} of plant-available N contains enough K to more than meet the need, leading to inorganic K input being zero in the intervention scenario. The direct soil emissions of N_2O are lower when manure replaces inorganic N fertiliser since the emission factor for the organic fertiliser is substantially lower.

Different crops, having different yields and nutritional values, contribute unequally to the total yield in cereal units. They also have different fertiliser demands. Sugar beet stands out by contributing the most to the total yields. Actual yields of sugar beet are an order of magnitude larger by weight than for the other crops, which after conversion to cereal unit is still high. However, sugar beet also has higher emissions from harvesting and crop transport due to having much higher mass. Excluding soil emissions (which are not dependent on crop type), the GWP100 per functional unit is the highest for rapeseed, followed by wheat, sugar beet and barley.

Responses in yields and N inputs to SOC changes

The change in SOC differs significantly between the scenarios which causes the difference in carbon emissions, but also influences the use of fertilisers and the yields. Figure 4 shows the changes over time of SOC, yields and optimal N input for winter wheat. The results show that the relative changes in yields and N inputs over 20 years are relatively small. Here, the changes of yields and N inputs are shown only for wheat as an example, however changes for other crops are similar.

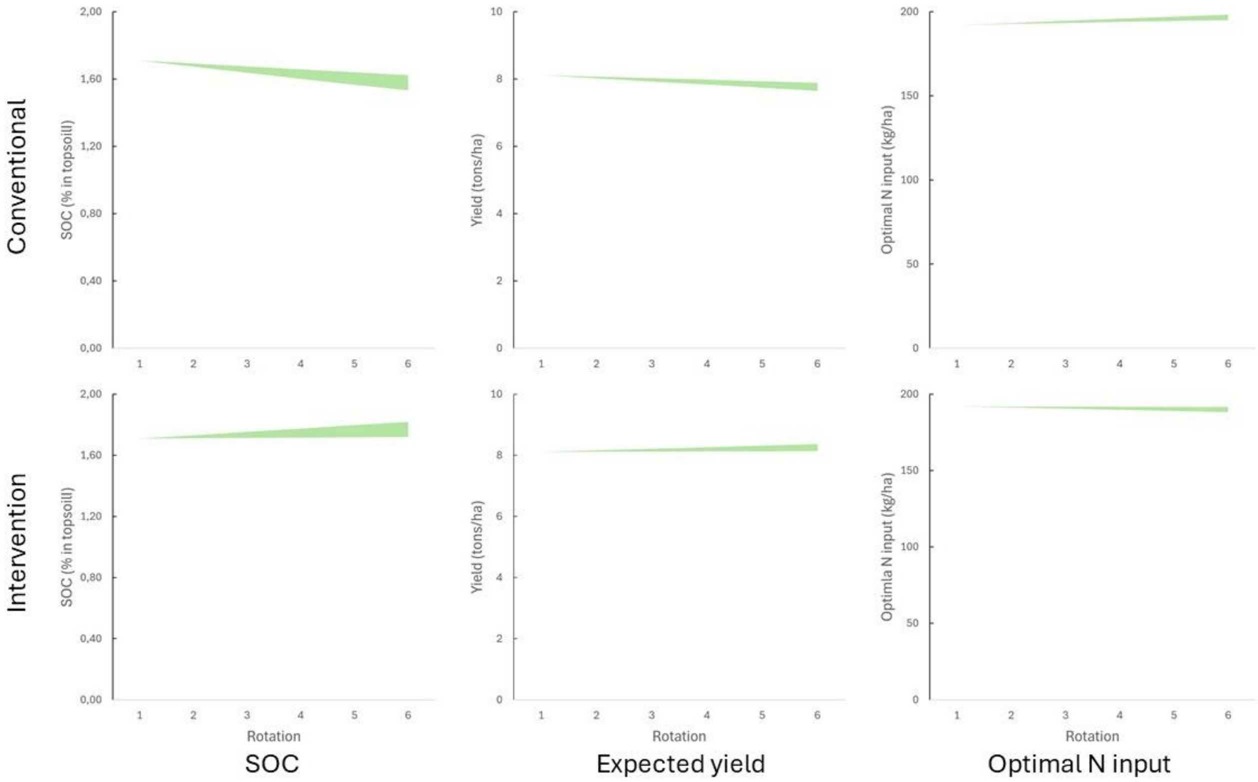

Figure 4. Development of SOC, expected yields and optimal N input for wheat from the base year and 20 years into the future. The 95 % confidence interval for the annual change rate is -0.405±0.145 % for the conventional scenario and 0.17±0.14 % for the intervention scenario (R. López i Losada, 2024, personal communication). Yields and optimal N input are calculated using production functions. Data points are taken every four years, when wheat occurs in the crop rotation.

Table 4
Comparison of SOC, yields and N inputs based on wheat between current average (rotation 1) and estimated values 20 years in the future (rotation 6). Based on mean values for SOC changes.

Scenario	SOC, conventional (% in topsoil)	Wheat yield, conventional (kg ha^{-1})	Optimal N input, conventional (kg ha^{-1})	SOC, intervention (% in topsoil)	Wheat yield, intervention (kg ha^{-1})	Optimal N input, intervention (kg ha^{-1})
Rotation 1	1,71	8111	192	1,71	8111	192
Rotation 6	1,58	7769	196,7	1,77	8252	189,9
Change	-7,8 %	-4,2 %	2,4 %	3,5 %	1,7 %	-1,1 %

SOC change rate margin of errors

To investigate whether possible scenarios within the margin of error for SOC changes (Figure 4) affect the conclusions drawn from the mean scenario (Figure 1), two alternatives where tested. A maximum SOC difference scenario was tested by assuming that the SOC change rate in the conventional scenario is at the upper bound of the confidence interval, while in the intervention scenario it is at the lower bound. The minimum SOC difference scenario assumes the opposite, that the conventional SOC change rate is at the lower bound and the intervention change rate is at the upper bound, maximizing the difference between scenarios. In the minimum difference scenario shown in Figure 5, the net total emissions of the intervention scenario are 31 % lower than in the conventional scenario. Excluding soil emissions of CO2 and N2O, the emissions from the management are 11 % lower in the intervention scenario.

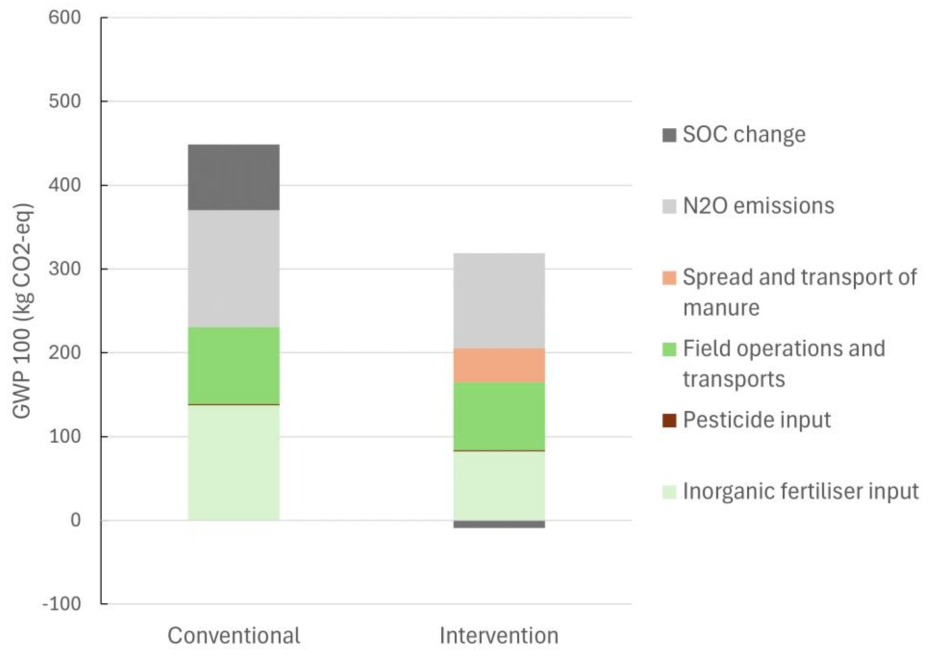

Figure 5 Total GWP100 per functional unit in the mininum SOC difference scenario. One full crop rotation, the last of the 20 year projection, is considered. The changes over time to SOC content, N input and yields derived from production functions are included. All inputs and processes described in table 3 are included. For SOC changes, upper bound values of the confidence interval are used in the conventional scenario while lower bound values are used in the intervention scenario.

In the maximum differnce scenario shown in Figure 6, the total emissions from the intervention scenario are 63 % lower than in the conventional scenario, excluding soil emissions the management emissions are 18 % lower. This comparison shows the effect that the margin of error of the SOC changes has on the results of the study.

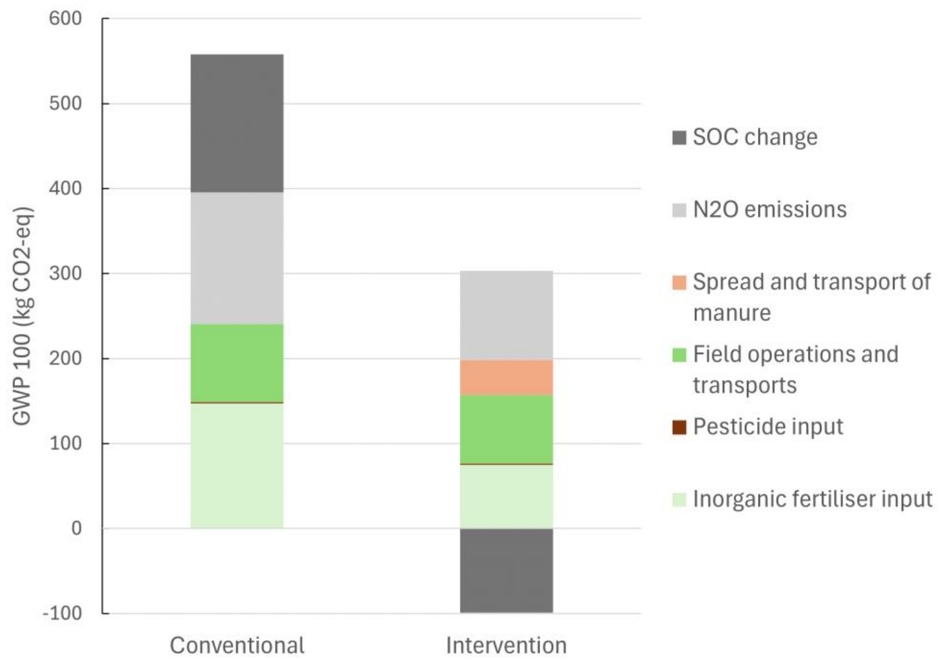

Figure 6 Total GWP100 per functional unit in the maximum SOC difference scenario. One full crop rotation, the last of the 20 year projection, is considered. The changes over time to SOC content, N input and yields derived from production functions are included. All inputs and processes described in table 3 are included. For SOC changes, lower bound values of the confidence interval are used in the conventional scenario while upper bound values are used in the intervention scenario.

Impact of GWP time horizon

The effects on the results of considering GWP from either a 20-year or 100-year perspective are minor. The direct soil emissions of CO_2 and N_2O are not considered different between 20- and 100-year perspectives. They are therefore not included in the comparison in Figure 7, which shows only emissions from inputs of fertilisers and pesticides, field operations and transports. The GWP20 is slightly higher than the GWP100 for every activity included, indicating some emissions of short lived but high potency greenhouse gases in the life cycle of each activity. From the production of inorganic fertilisers, the GWP20 is 8-9 % higher than GWP100. From field operations, it is 4-9 % higher while for pesticide production it is 17 % higher.

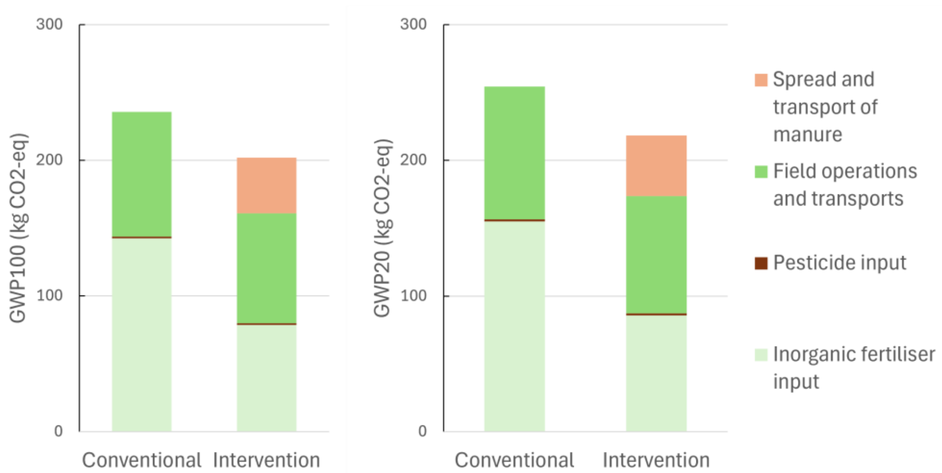

Figure 7 GWP 100 (left) compared to GWP 20 (right) of the management in each scenario. One full crop rotation, the last of the 20 year projection, is considered. The changes over time to SOC content, N input and yields derived from production functions are included. All inputs and processes described in table 1, except those related to soil emissions of CO_2 and N_2O, are included.

Discussion

The results of this study show a substantial difference in greenhouse gas emissions between the compared scenarios. The choice between a 20-year and 100-year perspective for GWP does not impact the results meaningfully, therefore only GWP100 is considered when discussing the results. The uncertainty of the SOC predictions can cause that difference to be smaller or bigger, but in any case, the intervention scenario performs better from a climate perspective. However, the difference in net emissions is built up mainly by the difference in SOC changes, while the emissions from the management are only slightly smaller in the intervention scenario. An explanation for this is that overall, the two scenarios are similar in terms of inputs and operations. In the intervention scenario, manure is used but most of the N is still supplied from inorganic fertiliser. The intervention scenario also has reduced tillage and lesser fuel consumption, but most of the field operations are the same. The SOC changes, however, differ significantly because the intervention scenario has a positive change, while the conventional scenario has a negative change. The relative difference of soil N_2O emissions between scenarios are significant, but these results are among the most uncertain, which will be further discussed in this section.

Management emissions

In this section, the reasons behind the differences in emissions from management, i.e. direct and indirect emissions from inputs, field operations and transports, and the effect of crop rotations are discussed. Both scenarios had the same crop rotation, however it is also apparent that different crops have different contributions to yields and fertiliser inputs (Figure 3). Different crops also have different pesticide inputs, and to some extent different field operations. This means that crops contribute differently to total GHG emissions, and alternative crop rotations could have higher or lower GWP.

Fertilisers and manure

One important factor contributing to the lesser emissions in the intervention scenario is the replacement of inorganic fertilisers with manure. The supply of manure is typically not allocated any environmental impact of the meat or dairy production it is sourced from. However, the GHG leakage during manure storage and transport can be allocated to the crop production in a LCA (Meisterling et al., 2009). During storage and transport of manure, greenhouse gases can be directly emitted from the manure (Hansen et al., 2006). In this study, no such emissions were considered before the manure was applied to the fields, leading to lower emissions for the intervention scenario compared to if direct manure emissions where quantified and allocated to the crop production. According to Hansen et al. (2006), a significant amount of the manure's carbon and nitrogen can be lost to the atmosphere during storage, reducing the positive effects of applying it to the fields. The loss of organic matter and nutrients, corresponding with the GHG emissions, can be reduced with the right handling of the manure, such as covering or straw content (Hansen et al., 2006; Sommer & Møller, 2000). In the comparison of scenarios in Figure 1, the soil N2O emissions are lower in the intervention scenario partially because the emission factor for organic N inputs is lower than for inorganic N. This difference could be negated by the manure emissions before field application. Hansen et al. (2006) report 4,8 % of initial nitrogen in uncovered manure being lost as N_2O, an order of magnitude higher than the 0,6 % assumed to be lost from manure applied in the field according to IPCC emission factors. How the emissions during storage should be allocated is another matter, as not applying manure to fields would not eliminate those emissions.

Decreasing the amount of manure in the intervention scenario increases the emissions from inorganic fertiliser production but decreases the emissions from manure transport and application. The net change is a very small decrease in in total emissions when manure inputs are decreased. This does not account for relationship between manure amount and carbon sequestration, which could be a more important factor in determining the optimal manure input. Applying large amounts of manure has been shown to increase SOC significantly in a short time, but this also increases N leakage (Blair et al., 2006). Additionally, as seen in Figure 2, the emissions related to manure are mainly those from the transportation, which assumes that manure is transported 60 km. 60 km is a rough estimation based on data from Agriwise and also assuming that local availability is low, requiring longer transports on avarage. The availability of manure varies, and emissions could be significantly lower, or higher, depending on proximity to animal agriculture.

Pesticides

The GHG emissions from pesticides were very small relative to other emission sources. There is no difference in the input of pesticides between scenarios, yet a small difference in emissions is present because of different values of the functional unit. The assumption that there would be no difference of inputs between scenarios might be false, given that tillage is an important factor in the management of weeds (Cordeau et al., 2020; Hobbs et al., 2008). Other studies have concluded that the exact effect of tillage on pesticide intensity is not clear (Deike et al., 2008; Sorensen et al., 2014). No method for quantifying the impact of reduced versus conventional tillage on pesticide use was identified in this study. Given the very low GHG emissions from pesticides in the current results, it is unlikely that it would affect the conclusions of the study.

Field operations and transport

This study shows that while GHG emissions from field operations are lower when tillage is reduced, the change is relatively small. The GWP100 from field operations, excluding transports, where 12 % lower in the intervention scenario. In another LCA study, Sorensen et al. (2014) found that reduced or no tillage uses respectively 26 % and 41 % less energy, corresponding to fuel use, compared to conventional tillage, indicating a slightly larger difference than in this study. This is likely an effect of different assumptions on tillage scenarios. Sorensen et al. (2014) also evaluated the effect of tillage on SOC and concluded that reducing the loss of SOC through reduced tillage had a bigger effect on the total GHG emissions than the reduced fuel use.

The GHG emissions from the agricultural management come primarily from the use of fossil fuels, either to extract and produce inorganic fertilisers or to power tractors and other machinery for field operations and transports. While soil carbon sequestration shows promise as a climate change mitigation measure, it will only partially compensate for the fossil fuel emissions in the world (Minasny et al., 2017; Schlesinger & Amundson, 2019). This thesis shows what while GHG emissions from management can be lowered by interventions that also improve SOC, the carbon sequestration is not enough to offset the remaining emissions. The conclusion is that the intervention scenario is performing better in terms of GHG emissions, but that emissions will need to be reduced further to reach climate neutrality. For example, the fossil fuel dependency of field operations must be dealt with in the future, e.g. through electrification (Lagnelöv et al., 2022).

Soil emissions

Soil organic carbon changes

The difference of SOC changes between scenarios is the most impactful aspect of the total difference in GHG emissions. The accuracy of the SOC change rates is therefore important for the robustness of the results. As concluded by Goglio et al. (2015), there are compromises between to be made in the method of accounting for SOC in LCA, regarding the robustness and accuracy of the results against the feasibility of the method. In this study, the data which the SOC change rates are based on is of high quality and specific to the climate of the region. The method of determining SOC changes is in line with what previous research has recommended (Haddaway et al., 2016; Sanderman & Baldock, 2010). This supports the accuracy and representativeness of the results. However, the SOC change rates are applicable to general categories of agricultural management, and do not describe in detail the effect of different crop rotations, having reduced tillage compared to no tillage, or the exact amount of manure being applied.

When assessing the ability of carbon sequestration to compensate for GHG emissions, time is an important aspect to consider. The permanence of carbon stocks in the soil is complex to determine and depend on many different factors (Smith, 2005). While the longevity of different GHG in the atmosphere is considered in the GWP metric for a certain period, the same cannot be said for SOC which complicates the comparison. It may be that soil carbon sequestration's primary potential lies in short term mitigation, only buying time to further reduce fossil emissions (Smith, 2012).

The assessment of the SOC change rate margin of error (Figure 3; Figure 4) shows that while the life cycle performance of the scenarios is affected, the difference between scenarios remains significant. Even if the carbon sequestration potential of the intervention scenario would be minimal, it has lower GHG emissions. However, the ability of the intervention management to achieve net negative emissions and act as a climate compensation measure can be questioned. Even in the best-case scenario for the intervention management (Figure 4), emissions are far from net zero.

Nitrous oxide emissions

The emissions of N_2O from soils were found to be significant in both scenarios and are also a major factor in the difference between scenarios. Like some previous LCA studies, it was found that soil N_2O is the single largest source of GHG emissions from crop production (Camargo et al., 2013; Nilsson et al., 2023). Other studies show varying results dependent on crop types and growing conditions (Aguilera et al., 2015;

Henryson et al., 2019). Direct and indirect soil emissions of nitrous oxide (N_2O) were calculated in this study using the IPCC approach, which roughly estimates emissions based on N inputs and a general characterization of climate (wet/dry) (Hergoualc'h et al., 2019). The interaction between N and C cycles in the soil are complicated, and effects are dependent on local soil and climatic conditions (Guenet et al., 2021; Wang et al., 2021). The IPCC tier 1 and 2 methodologies do not factor in all these local conditions and has been criticised for under-estimating N_2O emissions and having high uncertainty at the regional and local scale (Crutzen et al., 2008; Reay et al., 2012). The tier 2 approach is more reliable than the tier 1, but neither method considers local soil characteristics which can impact N_2O emissions (Goglio et al., 2018; Henryson et al., 2019).

The most important question, however, is whether there is a significant difference in soil N_2O emissions between the management scenarios. In a review of several meta-analyses, Guenet et al. (2021) find that measures to increase SOC can increase N_2O emissions which can partially off-set the carbon sequestration. Particularly, the adoption of reduced or no tillage can have a detrimental effect on N_2O emissions (Mei et al., 2018). However, another large meta-analysis found no significant effect on N_2O emissions from tillage intensity (van Kessel et al., 2013). On the contrast, the application of manure could have a positive effect, reducing the emissions of NH_3 which indirectly causes N_2O emissions (Xia et al., 2017). In summary, it is uncertain how the management scenarios in this study would differ in terms of N_2O emissions, since both scenarios includes methods that could have positive or negative effects. Future research should prioritize quantifying effects on N_2O emissions from management practices under regional conditions in GSS.

Limitations and future research

The use of LCA methodology means creating a model of a real system, based on data and assumptions. These assumptions may or may not have a significant impact on the results. Limitations of the method, such as what inputs and processes are included or not, may skew the results or affect the applicability of the results to other contexts. In this section, the potential effects of some assumptions and limitations in this study are discussed.

The ratio of N input and yields to SOC is dependent on crop prices and the costs of fertilisation. An increase of fertilisation costs of 50 % while crop prices remain the same would result in lower N input and yields, but only marginally reduce the GHG emissions per FU from fertilisation. The reduction in yields is also small, about 2 %, but might be a concern from a food security perspective. If crop prices increase while fertilisations costs stay the same, yields and emissions are still reduced. If crop prices change as much as the fertilisation costs, there is no change in yields or emissions. If fertilisation costs change, the cost of production would change in the same direction and logically, crop prices would roughly follow the same trend. The conclusion is that these costs and prices have little effect on the GHG emissions of agriculture.

The results of this study in terms of GHG emissions are put in relation to the functional unit, i.e. how much food is produced, meaning that higher yields will decrease the relative GWP. A fraction of the difference between the management scenarios can be explained by the slightly higher yields in the intervention scenario, which are expected to be higher over time because of the higher SOC content (Brady et al., 2015; Johnston et al., 2009). This is supported by the conclusions of Henryson et al. (2018), namely that increased yields from SOC contribute to the positive effects on the climate from increasing SOC, from an LCA perspective.

The used method of calculating yields does not consider other factors that can have an impact, which is significant if the management methods such as tillage affect yields in other ways. For example, the effect of reduced or no tillage on yields have previously been included in LCA (Sorensen et al., 2014). In a meta-analysis of 47 European studies, Van den Putte et al. (2010) found that reduced tillage can have a negative but limited effect on yields. Another, global, meta-analysis drew similar conclusions, that effects on yields are small especially in a temperate climate (Pittelkow et al., 2015). This implies that the relative GWP of the intervention scenario in this study could be underestimated, but not likely so much that the conclusions change.

The SOC change rate of the intervention scenario presumes that crop residues are left in the field. This had no consequences for the LCA and the GHG emissions, as it was not expected to significantly influence field operations. However, if system boundaries are expanded and functions other than food production are considered, the value of the crop residues that are harvested could affect the results. For example, straw from cereals and other residual products can be used for energy or as feed or bedding in animal production (Brankatschk & Finkbeiner, 2015; Gontard et al., 2018). These functions can be considered when calculating cereal units, and thus contribute to a higher environmental efficiency in the conventional scenario (Brankatschk & Finkbeiner, 2014). Future research could take a more holistic approach to assess the benefits and impacts of management in crop production.

Conclusions

The intervention scenario has significantly lower net GHG emissions than the conventional scenario. Considering the margin of error for SOC changes, this conclusion is still supported. The difference in SOC changes constituted the biggest portion of the total difference, followed by soil N_2O emissions and emissions from inorganic fertiliser inputs. Pesticide use caused very low GHG emissions relative to other inputs and processes. Transport and spread of manure in the intervention scenario negated some of the emission reduction. Management and emission allocation of manure may impact results since there can be significant losses of C and N from the manure before field application. Moreover, assumptions of transport distance for manure affect the results, indicating local availability of manure as an important factor.

The effect that differences in management has on direct and indirect N_2O emissions from the soil are not quantitatively addressed in this study. Some previous studies indicate that these effects are minor, while others argue that it negates the benefits of management promoting carbon sequestration. Given the scale of soil N_2O emissions, such effects could have significant impact on the results of the study, warranting further research.

Most of the difference between scenarios are in the soil emissions, while management emissions differ only slightly. This shows that the accuracy of methods to assess soil GHG emissions should be prioritised. In the intervention scenario, the carbon sequestration is far from offsetting all emissions, indicating that this form of intervention management is not sufficient to reach climate neutrality. Still, the results support the idea that intervention management is beneficial for the climate not only because of carbon sequestration, but because management and soil N_2O emissions are also lower.

Acknowledgements

Spending an entire semester on a single project has at times been a challenging process. It would not have been possible without certain people whose contributions I would like to acknowledge here. First, I would like to thank my supervisor Raül López i Losada, who has been actively present and supportive throughout the entire project, and who has cleared up my confusions one meeting or email at a time. I am also grateful to my other supervisor, professor Katarina Hedlund, for sharing her immense knowledge and providing insightful feedback at key points. Furthermore, thank you to Mark Brady for answer questions and providing data and materials, Martijn van Praagh for the rewarding group supervision sessions and to all my fellow thesis-writing master students for the chats, tips, and lunches. Lastly, I give my loving thanks to Sanna, who had to witness my least optimistic moments, and provided both advise and emotional support needed to carry on.

References

Abdalla, M., Hastings, A., Cheng, K., Yue, Q., Chadwick, D., Espenberg, M., Truu, J., Rees, R. M., & Smith, P. (2019). A critical review of the impacts of cover crops on nitrogen leaching, net greenhouse gas balance and crop productivity. *Global Change Biology*, *25*(8), 2530-2543. https://doi.org/10.1111/gcb.14644

AgriWise, 2020. *AgriWise: Data Book for Production Planning and Regional Enterprise Budgets*. Swedish Board of Agriculture.

Aguilera, E., Guzman, G., & Alonso, A. (2015). Greenhouse gas emissions from conventional and organic cropping systems in Spain. I. Herbaceous crops. *Agronomy for Sustainable Development*, *35*(2), 713-724. https://doi.org/10.1007/s13593-014-0267-9

Alvarez, R. (2005). A review of nitrogen fertilizer and conservation tillage effects on soil organic carbon storage. *Soil Use and Management*, *21*(1), 38-52. https://doi.org/10.1111/j.1475-2743.2005.tb00105.x

Bai, X., Huang, Y., Ren, W., Coyne, M., Jacinthe, P.-A., Tao, B., Hui, D., Yang, J., & Matocha, C. (2019). Responses of soil carbon sequestration to climate-smart agriculture practices: A meta-analysis. *Global Change Biology*, *25*(8), 2591-2606. https://doi.org/10.1111/gcb.14658

Baker, J. M., Ochsner, T. E., Venterea, R. T., & Griffis, T. J. (2007). Tillage and soil carbon sequestration—What do we really know? *Agriculture, Ecosystems & Environment*, *118*(1), 1-5. https://doi.org/10.1016/j.agee.2006.05.014

Barrios, E. (2007). Soil biota, ecosystem services and land productivity. *Ecological economics*, *64*(2), 269-285. https://doi.org/10.1016/j.ecolecon.2007.03.004

Bayer, C., Gomes, J., Vieira, F. C. B., Zanatta, J. A., de Cássia Piccolo, M., & Dieckow, J. (2012). Methane emission from soil under long-term no-till cropping systems. *Soil and Tillage Research*, *124*, 1-7. https://doi.org/10.1016/j.still.2012.03.006

Blair, N., Faulkner, R. D., Till, A. R., Korschens, M., & Schulz, E. (2006). Long-term management impacts on soil C, N and physical fertility: Part II: Bad Lauchstadt static and extreme FYM experiments. *Soil and Tillage Research*, *91*(1), 39-47. https://doi.org/10.1016/j.still.2005.11.001

Blanco-Canqui, H. (2021). No-till technology has limited potential to store carbon: How can we enhance such potential? *Agriculture, Ecosystems & Environment*, *313*, Article 107352. https://doi.org/10.1016/j.agee.2021.107352

Bolinder, M. A., Crotty, F., Elsen, A., Frac, M., Kismányoky, T., Lipiec, J., Tits, M., Tóth, Z., & Kätterer, T. (2020). The effect of crop residues, cover crops, manures and nitrogen fertilization on soil organic carbon changes in

agroecosystems: a synthesis of reviews. *Mitigation and Adaptation Strategies for Global Change*, *25*(6), 929-952. https://doi.org/10.1007/s11027-020-09916-3
Brady, M. V., Hedlund, K., Cong, R. G., Hemerik, L., Hotes, S., Machado, S., Mattsson, L., Schulz, E., & Thomsen, I. K. (2015). Valuing supporting soil ecosystem services in agriculture: a natural capital approach. *Agronomy Journal*, *107*(5), 1809-1821. https://doi.org/10.2134/agronj14.0597
Brady, M. V., Hristov, J., Wilhelmsson, F., & Hedlund, K. (2019). Roadmap for Valuing Soil Ecosystem Services to Inform Multi-Level Decision-Making in Agriculture. *Sustainability*, *11*(19), Article 5285. https://doi.org/10.3390/su11195285
Brandao, M., i Canals, L. M., & Clift, R. (2011). Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. *Biomass and bioenergy*, *35*(6), 2323-2336. https://doi.org/10.1016/j.biombioe.2009.10.019
Brankatschk, G., & Finkbeiner, M. (2014). Application of the Cereal Unit in a new allocation procedure for agricultural life cycle assessments. *Journal of Cleaner Production*, *73*, 72-79. https://doi.org/10.1016/j.jclepro.2014.02.005
Brankatschk, G., & Finkbeiner, M. (2015). Modeling crop rotation in agricultural LCAs — Challenges and potential solutions. *Agricultural Systems*, *138*, 66-76. https://doi.org/10.1016/j.agsy.2015.05.008
Camargo, G. G. T., Ryan, M. R., & Richard, T. L. (2013). Energy Use and Greenhouse Gas Emissions from Crop Production Using the Farm Energy Analysis Tool. *Bioscience*, *63*(4), 263-273. https://doi.org/10.1525/bio.2013.63.4.6
Cordeau, S., Baudron, A., & Adeux, G. (2020). Is Tillage a Suitable Option for Weed Management in Conservation Agriculture? *Agronomy*, *10*(11), Article 1746. https://doi.org/10.3390/agronomy10111746
Cowan, N., Maire, J., Krol, D., Cloy, J. M., Hargreaves, P., Murphy, R., Carswell, A., Jones, S. K., Hinton, N., & Anderson, M. (2021). Agricultural soils: A sink or source of methane across the British Isles? *European Journal of Soil Science*, *72*(4), 1842-1862. https://doi.org/10.1111/ejss.13075
Crutzen, P. J., Mosier, A. R., Smith, K. A., & Winiwarter, W. (2008). N2O release from agro-biofuel production negates global warming reduction by replacing fossil fuels. *Atmos. Chem. Phys.*, *8*(2), 389-395. https://doi.org/10.5194/acp-8-389-2008
Davidson, E. A., & Ackerman, I. L. (1993). Changes in soil carbon inventories following cultivation of previously untilled soils. *Biogeochemistry*, *20*(3), 161-193. https://doi.org/10.1007/BF00000786
Deike, S., Pallutt, B., Melander, B., Strassemeyer, J., & Christen, O. (2008). Long-term productivity and environmental effects of arable farming as affected by crop rotation, soil tillage intensity and strategy of pesticide use: A case-study of two long-term field experiments in Germany and Denmark. *European Journal of Agronomy*, *29*(4), 191-199. https://doi.org/10.1016/j.eja.2008.06.001
Ecoinvent. (2021). *ecoQuery*, version 3.8, allocation, classification by cut-off.

https://ecoquery.ecoinvent.org/3.8/cutoff/search
European Commission. (2023). *Pricing agricultural emissions and rewarding climate action in the agri-food value chain*. https://climate.ec.europa.eu/document/download/996c24d8-9004-4c4e-b637-60b384ae4814_en?filename=Pricing%20agricultural%20emissions%20and%20rewarding%20climate%20action%20in%20the%20agri-food%20value%20chain.pdf
Finnveden, G., Hauschild, M. Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., & Suh, S. (2009). Recent developments in Life Cycle Assessment. *Journal of Environmental Management*, *91*(1), 1-21. https://doi.org/10.1016/j.jenvman.2009.06.018
Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'Connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E. M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., . . . Zaks, D. P. M. (2011). Solutions for a cultivated planet. *Nature*, *478*(7369), 337-342. https://doi.org/10.1038/nature10452
Forster, P., Armour, K., Collins, W., Dufresne, J.-L., Frame, D., Lunt, D. J., Mauritsen, T., Palmer, M. D., Watanabe, M., Wild, M. & Zhang, H. (2021). In V. Masson-Delmotte, P. Zhai, A. Pirani, S. L. Connors, C. Péan, Y. Chen, L. Goldfarb, M. I. Gomis, J. B. Robin Matthews, S. Berger, M. Huang, O. Yelekci, R. Yu, B. Zhou, E. Lonnoy, T. K. Maycock, T. Waterfield, K. Lietzell, N. Caud (Eds.), *Climate Change 2021 The Physical Science Basis: Working Group I Contribution to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_Chapter07.pdf
Gardi, C., Jeffery, S., & Saltelli, A. (2013). An estimate of potential threats levels to soil biodiversity in EU. *Global Change Biology*, *19*(5), 1538-1548. https://doi.org/10.1111/gcb.12159
Goglio, P., Brankatschk, G., Knudsen, M. T., Williams, A. G., & Nemecek, T. (2018). Addressing crop interactions within cropping systems in LCA. *The International Journal of Life Cycle Assessment*, *23*(9), 1735-1743. https://doi.org/10.1007/s11367-017-1393-9
Goglio, P., Smith, W. N., Grant, B. B., Desjardins, R. L., Gao, X., Hanis, K., Tenuta, M., Campbell, C. A., McConkey, B. G., Nemecek, T., Burgess, P. J., & Williams, A. G. (2018). A comparison of methods to quantify greenhouse gas emissions of cropping systems in LCA. *Journal of Cleaner Production*, *172*, 4010-4017. https://doi.org/10.1016/j.jclepro.2017.03.133
Goglio, P., Smith, W. N., Grant, B. B., Desjardins, R. L., McConkey, B. G., Campbell, C. A., & Nemecek, T. (2015). Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. *Journal of Cleaner Production*, *104*, 23-39. https://doi.org/10.1016/j.jclepro.2015.05.040

Gontard, N., Sonesson, U., Birkved, M., Majone, M., Bolzonella, D., Celli, A., Angellier-Coussy, H., Jang, G.-W., Verniquet, A., & Broeze, J. (2018). A research challenge vision regarding management of agricultural waste in a circular bio-based economy. *Critical reviews in environmental science and technology*, *48*(6), 614-654. https://doi.org/10.1080/10643389.2018.1471957
Guenet, B., Gabrielle, B., Chenu, C., Arrouays, D., Balesdent, J. M., Bernoux, M., Bruni, E., Caliman, J. P., Cardinael, R., Chen, S. C., Ciais, P., Desbois, D., Fouche, J., Frank, S., Henault, C., Lugato, E., Naipal, V., Nesme, T., Obersteiner, M., . . . Zhou, F. (2021). Can N2O emissions offset the benefits from soil organic carbon storage? *Global Change Biology*, *27*(2), 237-256. https://doi.org/10.1111/gcb.15342
Gustafsson, H. & Johansson, C. (2008). *Reducerad jordbearbetning.* Jordbruksinformation 28–2008. Jordbruksverket. https://www2.jordbruksverket.se/webdav/files/SJV/trycksaker/Pdf_jo/jo08_28.pdf
Haddaway, N. R., Hedlund, K., Jackson, L. E., Kätterer, T., Lugato, E., Thomsen, I. K., Jørgensen, H. B., & Isberg, P.-E. (2016). Which agricultural management interventions are most influential on soil organic carbon (using time series data)? *Environmental Evidence*, *5*(1), Article 2. https://doi.org/10.1186/s13750-016-0053-z
Haddaway, N. R., Hedlund, K., Jackson, L. E., Kätterer, T., Lugato, E., Thomsen, I. K., Jørgensen, H. B., & Söderström, B. (2015). What are the effects of agricultural management on soil organic carbon in boreo-temperate systems? *Environmental Evidence*, *4*(1), Article 23. https://doi.org/10.1186/s13750-015-0049-0
Hansen, M. N., Henriksen, K., & Sommer, S. G. (2006). Observations of production and emission of greenhouse gases and ammonia during storage of solids separated from pig slurry: Effects of covering. *Atmospheric Environment*, *40*(22), 4172-4181. https://doi.org/10.1016/j.atmosenv.2006.02.013
Henryson, K., Hansson, E.-A., Katterer, T., Tidaker, P., & Sundberg, C. (2019). Environmental performance of crop cultivation at different sites and nitrogen rates in Sweden. *Nutrient Cycling in Agroecosystems*, *114*(2), 139-155. https://doi.org/10.1007/s10705-019-09997-w
Henryson, K., Sundberg, C., Katterer, T., & Hansson, P.-A. (2018). Accounting for long-term soil fertility effects when assessing the climate impact of crop cultivation. *Agricultural Systems*, *164*, 185-192. https://doi.org/10.1016/j.agsy.2018.03.001
Hergoualc'h, K., Akiyama, H., Bernoux, M., Chirinda, N., del Prado, A., Kasimir, Å., MacDonald, J. D., Ogle, S. M., Regina, K. & van der Weerden, T. J. (2019). Chapter 11: N2O emissions from managed soils, and CO2 emissions from lime and urea application. In E. Calvo Buenida, K. Tanabe, A. Kranjc, J. Baasansuren, M. Fukada, S. Ngarize, A. Osako, Y. Pyrozhenko, P. Shermanau & S. Federici (Eds.), *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 4: Agriculture, Forestry and Other Land Use*. IPCC. https://www.ipcc-

nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch11_Soils_N2O_CO2.pdf
Hobbs, P. R., Sayre, K., & Gupta, R. (2008). The role of conservation agriculture in sustainable agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *363*(1491), 543-555. https://doi.org/doi:10.1098/rstb.2007.2169
Huijbregts, M. A. J., Steinmann, Z. J. N., Elshout, P. M. F., Stam, G., Verones, F., Vieira, M., Zijp, M., Hollander, A., & van Zelm, R. (2017). ReCiPe2016: a harmonised life cycle impact assessment method at midpoint and endpoint level. *The International Journal of Life Cycle Assessment*, *22*(2), 138-147. https://doi.org/10.1007/s11367-016-1246-y
IPCC. (2019). Summary for Policymakers. In P.R. Shukla, J. Skea, E. Calvo Buendia, V. Masson-Delmotte, H.- O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. Portugal Pereira, P. Vyas, E. Huntley, K. Kissick, M. Belkacemi, J. Malley, (Eds.), *Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems.* https://www.ipcc.ch/site/assets/uploads/sites/4/2022/11/SRCCL_SPM.pdf
IPCC, 2023: Summary for Policymakers. In H. Lee, K. Calvin, D. Dasgupta, G. Krinner, A. Mukherji, P. Thorne, C. Trisos, J. Romero, P. Aldunce, K. Barrett, G, Blanco, W. W. L. Cheung, S. L. Connors, F. Denton, A. Diongue-Niang, D. Dodman, M. Garschagen, O. Geden, B. Hayward, … Z. Zommers, *Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment Report of the Intergovernmental Panel on Climate*. https://doi.org/10.59327/IPCC/AR6-9789291691647.001
Joensuu, K., Rimhanen, K., Heusala, H., Saarinen, M., Usva, K., Leinonen, I., & Palosuo, T. (2021). Challenges in using soil carbon modelling in LCA of agricultural products—the devil is in the detail. *The International Journal of Life Cycle Assessment*, *26*(9), 1764-1778. https://doi.org/10.1007/s11367-021-01967-1
Johnston, A. E., Poulton, P. R., & Coleman, K. (2009). Chapter 1 Soil Organic Matter: Its Importance in Sustainable Agriculture and Carbon Dioxide Fluxes. *Advances in Agronomy*, *101*, 1-57. https://doi.org/10.1016/S0065-2113(08)00801-8
Jordbruksverket. (2020). Bilaga 2: Områdesindelningar i jordbruksstatistiken. In Jordbruksverket, *Jordbruksstatistisk sammanställning 2020*. https://jordbruksverket.se/download/18.78dd5d7d173e2fbbcda98885/1597390149090/23_Bilaga%202_2020.pdf
Khan, S. A., Mulvaney, R. L., Ellsworth, T. R., & Boast, C. W. (2007). The Myth of Nitrogen Fertilization for Soil Carbon Sequestration. *Journal of Environmental Quality*, *36*(6), 1821-1832. https://doi.org/10.2134/jeq2007.0099

Kottek, M., Grieser, J., Beck, C., Rudolf, B., & Rubel, F. (2006). World map of the Köppen-Geiger climate classification updated. *Meteorologische Zeitschrift*, *15*(3), 269-263. https://doi.org/10.1127/0941-2948/2006/0130

Lagnelöv, O., Larsson, G., Larsolle, A. & Hansson, P.-A. (2022). *El-traktorers potential att minska Sveriges klimatpåverkan: En studie av maskinsystem i lantbruket*. Mistra Food Futures Report 10. https://mistrafoodfutures.se/wp-content/uploads/2022/11/mistra-food-futures-report-10-web.pdf

Lal, R. (2004). Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. *Science*, *304*(5677), 1623-1627. https://doi.org/doi:10.1126/science.1097396

Le Mer, J., & Roger, P. (2001). Production, oxidation, emission and consumption of methane by soils: A review. *European Journal of Soil Biology*, *37*(1), 25-50. https://doi.org/10.1016/S1164-5563(01)01067-6

Ludwig, B., Geisseler, D., Michel, K., Joergensen, R. G., Schulz, E., Merbach, I., Raupp, J., Rauber, R., Hu, K., Niu, L., & Liu, X. (2011). Effects of fertilization and soil management on crop yields and carbon stabilization in soils. A review. *Agronomy for Sustainable Development*, *31*(2), 361-372. https://doi.org/10.1051/agro/2010030

Mei, K., Wang, Z., Huang, H., Zhang, C., Shang, X., Dahlgren, R. A., Zhang, M., & Xia, F. (2018). Stimulation of N2O emission by conservation tillage management in agricultural lands: A meta-analysis. *Soil and Tillage Research*, *182*, 86-93. https://doi.org/10.1016/j.still.2018.05.006

Meisterling, K., Samaras, C., & Schweizer, V. (2009). Decisions to reduce greenhouse gases from agriculture and product transport: LCA case study of organic and conventional wheat. *Journal of Cleaner Production*, *17*(2), 222-230. https://doi.org/10.1016/j.jclepro.2008.04.009

Menegat, S., Ledo, A., & Tirado, R. (2022). Greenhouse gas emissions from global production and use of nitrogen synthetic fertilisers in agriculture. *Scientific Reports*, *12*(1), Article 14490. https://doi.org/10.1038/s41598-022-18773-w

Minasny, B., Malone, B. P., McBratney, A. B., Angers, D. A., Arrouays, D., Chambers, A., Chaplot, V., Chen, Z.-S., Cheng, K., Das, B. S., Field, D. J., Gimona, A., Hedley, C. B., Hong, S. Y., Mandal, B., Marchant, B. P., Martin, M., McConkey, B. G., Mulder, V. L., . . . Winowiecki, L. (2017). Soil carbon 4 per mille. *Geoderma*, *292*, 59-86. https://doi.org/10.1016/j.geoderma.2017.01.002

Nemecek, T., Dubois, D., Huguenin-Elie, O., & Gaillard, G. (2011). Life cycle assessment of Swiss farming systems: I. Integrated and organic farming. *Agricultural Systems*, *104*(3), 217-232. https://doi.org/10.1016/j.agsy.2010.10.002

Nemecek, T. & Kägi, T. (2007). *Life cycle inventories of agricultural production systems: Data v2.0*. Ecoinvent report No. 15. https://db.ecoinvent.org/reports/15_Agriculture.pdf

Nilsson, J., El Khosht, F. F., Bergkvist, G., Öborn, I., & Tidåker, P. (2023). Effect of short-term perennial leys on life cycle environmental performance of cropping systems: An assessment based on data from a long-term field

experiment. *European Journal of Agronomy*, *149*, 126888. https://doi.org/10.1016/j.eja.2023.126888
Oldfield, E. E., Bradford, M. A., & Wood, S. A. (2019). Global meta-analysis of the relationship between soil organic matter and crop yields. *SOIL*, *5*(1), 15-32. https://doi.org/10.5194/soil-5-15-2019
Pan, B., Lam, S. K., Mosier, A., Luo, Y., & Chen, D. (2016). Ammonia volatilization from synthetic fertilizers and its mitigation strategies: A global synthesis. *Agriculture Ecosystems & Environment*, *232*, 283-289. https://doi.org/10.1016/j.agee.2016.08.019
Phillips, R. E., Blevins, R. L., Thomas, G. W., Frye, W. W., & Phillips, S. H. (1980). No-Tillage Agriculture. *Science*, *208*(4448), 1108-1113. http://www.jstor.org/stable/1683700
Piikki, K., & Söderström, M. (2019). Digital soil mapping of arable land in Sweden – Validation of performance at multiple scales. *Geoderma*, *352*, 342-350. https://doi.org/10.1016/j.geoderma.2017.10.049
Pittelkow, C. M., Linquist, B. A., Lundy, M. E., Liang, X., van Groenigen, K. J., Lee, J., van Gestel, N., Six, J., Venterea, R. T., & van Kessel, C. (2015). When does no-till yield more? A global meta-analysis. *Field Crops Research*, *183*, 156-168. https://doi.org/10.1016/j.fcr.2015.07.020
Powlson, D., Goulding, K., Willison, T. W., Webster, C., & Hütsch, B. (1997). The effect of agriculture on methane oxidation in soil. *Nutrient Cycling in Agroecosystems*, *49*, 59-70. https://doi.org/10.1023/A:1009704226554
Prechsl, U. E., Wittwer, R., van der Heijden, M. G. A., Lüscher, G., Jeanneret, P., & Nemecek, T. (2017). Assessing the environmental impacts of cropping systems and cover crops: Life cycle assessment of FAST, a long-term arable farming field experiment. *Agricultural Systems*, *157*, 39-50. https://doi.org/10.1016/j.agsy.2017.06.011
Reay, D. S., Davidson, E. A., Smith, K. A., Smith, P., Melillo, J. M., Dentener, F., & Crutzen, P. J. (2012). Global agriculture and nitrous oxide emissions. *Nature Climate Change*, *2*(6), 410-416. https://doi.org/10.1038/nclimate1458
Sanderman, J., & Baldock, J. A. (2010). Accounting for soil carbon sequestration in national inventories: a soil scientist's perspective. *Environmental Research Letters*, *5*(3), Article 034003. https://doi.org/10.1088/1748-9326/5/3/034003
SCB. (2022). *Växtskyddsmedel i jord- och trädgårdsbruket 2021, Användning i grödor.* https://www.scb.se/contentassets/6e042f0902bb449fb15edc4c1eb8e22c/mi0502_2021i20_br_mi31br2202.pdf
SCB. (2023). *Gödselmedel i jordbruket 2021/22 Mineral- och stallgödsel till olika grödor samt hantering och lagring av stallgödsel.* https://www.scb.se/contentassets/6d115170251e40149815b2e101e5e396/mi1001_2021b22_sm_mi30sm2302.pdf
Schlesinger, W. H., & Amundson, R. (2019). Managing for soil carbon sequestration: Let's get realistic. *Global Change Biology*, *25*(2), 386-389. https://doi.org/10.1111/gcb.14478

Schlesinger, W. H., & Bernhardt, E. S. (2020). Chapter 11 - The Global Carbon and Oxygen Cycles. In W. H. Schlesinger & E. S. Bernhardt (Eds.), *Biogeochemistry* (4th ed, pp. 453-481). Academic Press. https://doi.org/10.1016/B978-0-12-814608-8.00011-6

Smith, P. (2005). An overview of the permanence of soil organic carbon stocks: influence of direct human-induced, indirect and natural effects. *European Journal of Soil Science*, *56*(5), 673-680. https://doi.org/10.1111/j.1365-2389.2005.00708.x

Smith, P. (2012). Soils and climate change. *Current Opinion in Environmental Sustainability*, *4*(5), 539-544. https://doi.org/10.1016/j.cosust.2012.06.005

Sommer, S. G., & Møller, H. B. (2000). Emission of greenhouse gases during composting of deep litter from pig production – effect of straw content. *The Journal of Agricultural Science*, *134*(3), 327-335. https://doi.org/10.1017/S0021859699007625

Sorensen, C. G., Halberg, N., Oudshoorn, F. W., Petersen, B. M., & Dalgaard, R. (2014). Energy inputs and GHG emissions of tillage systems. *Biosystems Engineering*, *120*, 2-14. https://doi.org/10.1016/j.biosystemseng.2014.01.004

Svenska institutet för standarder. (2006a). *Miljöledning: Livscykelanalys: Principer och struktur* (SS-EN ISO 14040:2006). https://www.sis.se/produkter/miljo-och-halsoskydd-sakerhet/miljoskydd/sseniso14040/

Svenska institutet för standarder. (2006b). *Miljöledning: Livscykelanalys: Krav och vägledning* (SS-EN ISO 14044:2006). https://www.sis.se/produkter/ledningssystem-e07b0fe8/ledningssystem-for-miljo/sseniso140442006/

Tillman, A.-M. (2000). Significance of decision-making for LCA methodology. *Environmental Impact Assessment Review*, *20*(1), 113-123. https://doi.org/10.1016/S0195-9255(99)00035-9

Tsiafouli, M. A., Thébault, E., Sgardelis, S. P., de Ruiter, P. C., van der Putten, W. H., Birkhofer, K., Hemerik, L., de Vries, F. T., Bardgett, R. D., Brady, M. V., Bjornlund, L., Jørgensen, H. B., Christensen, S., Hertefeldt, T. D., Hotes, S., Gera Hol, W. H., Frouz, J., Liiri, M., Mortimer, S. R., . . . Hedlund, K. (2015). Intensive agriculture reduces soil biodiversity across Europe. *Global Change Biology*, *21*(2), 973-985. https://doi.org/10.1111/gcb.12752

Tubiello, F. N., Salvatore, M., Ferrara, A. F., House, J., Federici, S., Rossi, S., Biancalani, R., Condor Golec, R. D., Jacobs, H., Flammini, A., Prosperi, P., Cardenas-Galindo, P., Schmidhuber, J., Sanz Sanchez, M. J., Srivastava, N., & Smith, P. (2015). The Contribution of Agriculture, Forestry and other Land Use activities to Global Warming, 1990–2012. *Global Change Biology*, *21*(7), 2655-2660. https://doi.org/10.1111/gcb.12865

Van den Putte, A., Govers, G., Diels, J., Gillijns, K., & Demuzere, M. (2010). Assessing the effect of soil tillage on crop growth: A meta-regression analysis on European crop yields under conservation agriculture. *European Journal of Agronomy*, *33*(3), 231-241. https://doi.org/10.1016/j.eja.2010.05.008

van Kessel, C., Venterea, R., Six, J., Adviento-Borbe, M. A., Linquist, B., & van Groenigen, K. J. (2013). Climate, duration, and N placement determine N2O emissions in reduced tillage systems: a meta-analysis. *Global Change Biology*, *19*(1), 33-44. https://doi.org/10.1111/j.1365-2486.2012.02779.x
VandenBygaart, A. J. (2016). The myth that no-till can mitigate global climate change. *Agriculture, Ecosystems & Environment*, *216*, 98-99. https://doi.org/10.1016/j.agee.2015.09.013
Wang, C., Amon, B., Schulz, K., & Mehdi, B. (2021). Factors That Influence Nitrous Oxide Emissions from Agricultural Soils as Well as Their Representation in Simulation Models: A Review. *Agronomy-Basel*, *11*(4), Article 770. https://doi.org/10.3390/agronomy11040770
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment*, *21*(9), 1218-1230. https://doi.org/10.1007/s11367-016-1087-8
Xia, L., Lam, S. K., Yan, X., & Chen, D. (2017). How Does Recycling of Livestock Manure in Agroecosystems Affect Crop Productivity, Reactive Nitrogen Losses, and Soil Carbon Balance?. *Environmental Science & Technology*, *51*(13), 7450-7457. https://doi.org/10.1021/acs.est.6b06470
Zomer, R. J., Bossio, D. A., Sommer, R., & Verchot, L. V. (2017). Global Sequestration Potential of Increased Organic Carbon in Cropland Soils. *Scientific Reports*, *7*(1), Article 15554. https://doi.org/10.1038/s41598-017-15794-8

Appendix

Here, supplementary data explaining some of the details of the method are presented. Calculations of yields and fertiliser inputs as a function of SOC, using production functions of Brady et al. (2019, were based on the numbers presented in table A1. Tables A3-A6 detail how SOC, yields, and fertiliser inputs as well as GHG emissions from different categories change over time and between different crops. GHG emissions are presented per hectare, and not per functional unit. However, only the last full crop rotation is considered when presenting GHG emissions in the results of the thesis.

Table A1
Typical inputs and yields conditions for different crops in GSS, based on AgriWise (2020) data. Fertilisation costs includes the costs of P and K fertiliser as well as energy costs from fertiliser application, explaining the different values for different crops.

	W. wheat	S. barley	Rapeseed	Sugar beet	Reference
Yield (kg ha^{-1})	8111	6273	3625	71943	Agriwise (2020)
N input (kg N ha^{-1})	192	108	173	120	Agriwise (2020)
P input (kg P kg N^{-1})	0,125	0,1734	0,1503	0,2667	Agriwise (2020)
K input (kg K kg N^{-1})	0,1354	0,1468	0,1503	0,1333	Agriwise (2020)
Crop price (SEK kg^{-1})	1,51	1,45	3,64	0,32	Brady et al. (2019)
Fertilisation cost (SEK kg N^{-1})	16,0	15,1	20,8	33,5	Brady et al. (2019)
Pesticide input (kg active substance ha^{-1})	1,41	0,70	0,44	2,38	SCB (2022)

Table A2

List of ecoinvent activities used in the model, the geographic scope of the ecoinvent data and the source data used to quantify inputs in this study. These activities represent all greenhouse gas emissions included in the LCA except for SOC changes and soil N_2O emissions.

Activity	Ecoinvent geography	Input data
Inorganic N fertiliser	Sweden	See table A1
Inorganic P fertiliser	Sweden	See table A1
Inorganic K fertiliser	Sweden	See table A1
Packaging for fertiliser	Global	Based on N/P/K concentration in fertilisers representing >95 % of Swedish market according to ecoinvent
Pesticide, unspecified	Global	See table A1
Tillage, rolling	Switzerland	Adapted from Gustafsson & Johansson (2008)
Tillage, Harrowing, By Rotary Harrow	Switzerland	Adapted from Gustafsson & Johansson (2008)
Tillage, Harrowing, By Spring Tine Harrow	Switzerland	Adapted from Gustafsson & Johansson (2008)
Tillage, Ploughing	Switzerland	Adapted from Gustafsson & Johansson (2008)
Application Of Plant Protection Product, By Field Sprayer	Switzerland	Nilsson et al. (2023)
Fertilising, By Broadcaster	Switzerland	Nilsson et al. (2023)
Combine harvesting	Switzerland	ecoinvent (2021)
Harvesting, beets	Switzerland	ecoinvent (2021)
Sowing	Switzerland	Adapted from Gustafsson & Johansson (2008)
Solid manure loading and spreading, by hydraulic loader and spreader	Switzerland	Function of amount of solid manure used, based on SCB (2023)
Liquid manure spreading, by vacuum tanker	Switzerland	Function of amount of liquid manure used, based on SCB (2023)
Transport, tractor and trailer	Switzerland	Based on crop yields assuming 30 km transport (Agriwise, 2020), plus amount of manure assuming 60 km transport

Table A3
SOC, yields and nutrient inputs in the conventional scenario, modelled 20 years into the future using production functions. Mean values for SOC change rates are used.

Year	Crop	SOC mean (% in 30 cm topsoil)	Yield (kg ha^{-1})	Cereal unit (kg ha^{-1})	Optimal N input (kg ha^{-1})	Optimal P input (kg ha^{-1})	Optimal K input (kg ha^{-1})
0	Wheat	1,71	8111	8217	192,00	24,00	26,00
1	Rapeseed	1,70	3606	4680	173,00	26,00	26,00
2	Barley	1,70	6262	6082	109,22	18,94	16,03
3	Sugar beet	1,69	71692	16477	125,02	33,34	16,67
4	Wheat	1,68	8043	8146	192,97	24,12	26,13
5	Rapeseed	1,68	3528	4579	173,00	26,00	26,00
6	Barley	1,67	6240	6060	109,66	19,02	16,10
7	Sugar beet	1,66	71342	16397	131,62	35,10	17,54
8	Wheat	1,66	7975	8075	193,92	24,24	26,26
9	Rapeseed	1,65	3451	4478	173,00	26,00	26,00
10	Barley	1,64	6217	6037	110,09	19,09	16,16
11	Sugar beet	1,64	70974	16312	138,11	36,83	18,41
12	Wheat	1,63	7906	8004	194,86	24,36	26,38
13	Rapeseed	1,62	3373	4378	173,00	26,00	26,00
14	Barley	1,62	6194	6014	110,52	19,16	16,22
15	Sugar beet	1,61	70589	16223	144,50	38,54	19,26
16	Wheat	1,60	7838	7933	195,79	24,47	26,51
17	Rapeseed	1,60	3296	4277	173,00	26,00	26,00
18	Barley	1,59	6171	5991	110,94	19,24	16,29
19	Sugar beet	1,58	70188	16131	150,79	40,21	20,10
20	Wheat	1,58	7769	7861	196,70	24,59	26,63

Table A4
GWP (100 years) of different categories for each year in the conventional scenario. The category inorganic fertilisers include the production of N, P and K fertilisers including transport to the customers. Pesticides includes production and transport. Field operations include sowing, tillage, application of pesticides and inorganic fertilisers and harvesting.

Year	**Crop**	**SOC change (kg CO_2-eq ha^{1-})**	**N2O emissions (kg CO_2-eq ha^{1-})**	**GWP100 inorganic fertilisers (kg CO_2-eq ha^{1-})**	**GWP100 pesticides (kg CO_2-eq ha^{1-})**	**GWP 100 fertiliser packaging (kg CO_2-eq ha^{1-})**	**GWP 100 crop tranport (kg CO_2-eq ha^{1-})**	**GWP 100 manure transport (kg CO_2-eq ha^{1-})**	**GWP 100 manure spread (kg CO_2-eq ha^{1-})**	**GWP 100 field operations (kg CO_2-eq ha^{1-})**
0	Wheat	0,00	1444,73	1322,73	13,42	94,97	87,22	0,00	0,00	408,20
1	Rapeseed	1112,11	1379,84	1233,46	4,19	86,18	38,78	0,00	0,00	399,29
2	Barley	1107,62	899,62	792,20	6,66	54,65	67,34	0,00	0,00	242,42
3	Sugar beet	1103,14	1018,17	969,47	22,66	63,65	770,97	0,00	0,00	1155,07
4	Wheat	1098,68	1529,16	1329,42	13,42	95,45	86,49	0,00	0,00	408,20
5	Rapeseed	1094,24	1378,58	1233,46	4,19	86,18	37,94	0,00	0,00	399,29
6	Barley	1089,82	901,68	795,39	6,66	54,87	67,10	0,00	0,00	242,42
7	Sugar beet	1085,42	1066,59	1020,64	22,66	67,01	767,20	0,00	0,00	1155,07
8	Wheat	1081,03	1535,10	1335,99	13,42	95,92	85,76	0,00	0,00	408,20
9	Rapeseed	1076,66	1377,35	1233,46	4,19	86,18	37,11	0,00	0,00	399,29
10	Barley	1072,31	903,70	798,53	6,66	55,08	66,86	0,00	0,00	242,42
11	Sugar beet	1067,97	1114,22	1070,99	22,66	70,32	763,24	0,00	0,00	1155,07
12	Wheat	1063,66	1540,95	1342,46	13,42	96,39	85,02	0,00	0,00	408,20
13	Rapeseed	1059,36	1376,13	1233,46	4,19	86,18	36,28	0,00	0,00	399,29
14	Barley	1055,08	905,69	801,61	6,66	55,30	66,61	0,00	0,00	242,42
15	Sugar beet	1050,81	1161,09	1120,54	22,66	73,57	759,09	0,00	0,00	1155,07
16	Wheat	1046,57	1546,70	1348,83	13,42	96,84	84,28	0,00	0,00	408,20
17	Rapeseed	1042,34	1374,94	1233,46	4,19	86,18	35,44	0,00	0,00	399,29
18	Barley	1038,12	907,65	804,65	6,66	55,50	66,36	0,00	0,00	242,42
19	Sugar beet	1033,93	1207,20	1169,28	22,66	76,77	754,78	0,00	0,00	1155,07
20	Wheat	1029,75	1552,36	1355,09	13,42	97,29	83,54	0,00	0,00	408,20

Table A5

SOC, yields and nutrient inputs in the intervention scenario, modelled 20 years into the future using production functions. Mean values for SOC change rates are used.

Year	Crop	SOC mean (% in 30 cm topsoil)	Yield (kg ha^{-1})	Cereal unit (kg ha^{-1})	Optimal N input (kg ha^{-1})	Optimal P input (kg ha^{-1})	Optimal K input (kg ha^{-1})
0	Wheat	1,71	8111	8217	192,00	24,00	26,00
1	Rapeseed	1,71	3633	4715	173,00	26,00	26,00
2	Barley	1,72	6278	6098	108,91	18,88	15,99
3	Sugar beet	1,72	72045	16558	117,87	31,44	15,71
4	Wheat	1,72	8139	8247	191,59	23,95	25,94
5	Rapeseed	1,72	3665	4757	173,00	26,00	26,00
6	Barley	1,73	6287	6107	108,72	18,85	15,96
7	Sugar beet	1,73	72178	16589	115,02	30,68	15,33
8	Wheat	1,73	8168	8276	191,17	23,90	25,88
9	Rapeseed	1,74	3698	4799	173,00	26,00	26,00
10	Barley	1,74	6296	6116	108,53	18,82	15,93
11	Sugar beet	1,74	72307	16619	112,15	29,91	14,95
12	Wheat	1,75	8196	8305	190,76	23,84	25,83
13	Rapeseed	1,75	3730	4841	173,00	26,00	26,00
14	Barley	1,75	6304	6124	108,34	18,79	15,90
15	Sugar beet	1,75	72432	16647	109,26	29,14	14,56
16	Wheat	1,76	8224	8335	190,34	23,79	25,77
17	Rapeseed	1,76	3762	4883	173,00	26,00	26,00
18	Barley	1,76	6313	6133	108,14	18,75	15,88
19	Sugar beet	1,77	72554	16675	106,35	28,36	14,18
20	Wheat	1,77	8252	8364	189,91	23,74	25,71

Table A6

GWP (100 years) of different categories for each year in the intervention scenario. The category inorganic fertilisers include the production of N, P and K fertilisers including transport to the customers. Pesticides includes production and transport. Field operations include sowing, tillage, application of pesticides and inorganic fertilisers and harvesting.

Year	Crop	Yearly soil emission CO2-eq (kg)	N2O emissions (kg CO2-eq)	GWP100 Fert.	GWP100 Pesticides GSS	GWP 100 packaging fert.	Crop transport GWP100	Manure transport GWP100	Manure spread GWP100	GWP 100 field operations
0	Wheat	0,00	1345,73	907,50	13,42	77,40	87,22	347,23	21,82	344,90
1	Rapeseed	-468,16	1201,39	818,20	4,19	68,61	39,07	347,23	21,82	335,35
2	Barley	-468,95	714,49	431,09	6,66	37,52	67,51	347,23	21,82	169,54
3	Sugar beet	-469,75	782,62	556,80	22,66	43,06	774,76	347,23	21,82	1094,76
4	Wheat	-470,55	1342,60	904,98	13,42	77,20	87,53	347,23	21,82	344,90
5	Rapeseed	-471,35	1201,39	818,20	4,19	68,61	39,42	347,23	21,82	335,35
6	Barley	-472,15	713,05	429,88	6,66	37,42	67,60	347,23	21,82	169,54
7	Sugar beet	-472,96	760,96	536,83	22,66	41,63	776,18	347,23	21,82	1094,76
8	Wheat	-473,76	1339,45	902,44	13,42	77,00	87,83	347,23	21,82	344,90
9	Rapeseed	-474,57	1201,39	818,20	4,19	68,61	39,77	347,23	21,82	335,35
10	Barley	-475,38	711,61	428,66	6,66	37,33	67,70	347,23	21,82	169,54
11	Sugar beet	-476,18	739,16	516,73	22,66	40,19	777,57	347,23	21,82	1094,76
12	Wheat	-476,99	1336,28	899,88	13,42	76,79	88,14	347,23	21,82	344,90
13	Rapeseed	-477,81	1201,39	818,20	4,19	68,61	40,11	347,23	21,82	335,35
14	Barley	-478,62	710,15	427,43	6,66	37,24	67,80	347,23	21,82	169,54
15	Sugar beet	-479,43	717,20	496,49	22,66	38,74	778,92	347,23	21,82	1094,76
16	Wheat	-480,25	1333,08	897,31	13,42	76,59	88,44	347,23	21,82	344,90
17	Rapeseed	-481,07	1201,39	818,20	4,19	68,61	40,46	347,23	21,82	335,35
18	Barley	-481,88	708,69	426,19	6,66	37,14	67,89	347,23	21,82	169,54
19	Sugar beet	-482,70	695,10	476,12	22,66	37,29	780,23	347,23	21,82	1094,76
20	Wheat	-483,53	1329,87	894,71	13,42	76,38	88,74	347,23	21,82	344,90

LUNDS UNIVERSITET

WWW.CEC.LU.SE
WWW.LU.SE

Lunds universitet

Miljövetenskaplig utbildning
Centrum för miljö- och
klimatvetenskap
Ekologihuset
223 62 Lund